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Effect of vehicle characteristics on unpaved road dust emissions

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Abstract

This paper presents PM_{10} fugitive dust emission factors for a range of vehicles types and examines the influence of vehicle and wake characteristics on the strength of emissions from an unpaved road. Vertical profile measurements of mass concentration of the passing plumes were carried out using a series of 3 instrumented towers. PM_{10} emission fluxes at each tower were calculated from knowledge of the vertical mass concentration profile, the ambient wind speed and direction, and the time the plume took to pass the towers. The emission factors showed a strong linear dependence on speed and vehicle weight. Emission factors (EF = grams of PM_{10} emitted per vehicle kilometer traveled) ranged from approximately $EF = 0.8 \times (\mathrm{km\,h^{-1}})$ for a light (~1200 kg) passenger car to $EF = 48 \times (\mathrm{km\,h^{-1}})$ for large military vehicles (~18 000 kg). In comparison to emission estimates derived using US EPA AP-42 methods the measured emission factors indicate larger than estimated contributions for speeds generally > 10–20 km h^{-1} and for vehicle weights > 3000 kg. The size of a wake created by a vehicle was observed to be dependent on the size of the vehicle, increasing roughly linearly with vehicle height. Injection height of the dust plume is least important to long-range transport of PM_{10} under unstable conditions and most important under stable atmospheric conditions.

Keyword: Fugitive dust; PM_{10}; Unpaved road dust emissions; Emission factors

1. Introduction

Most unpaved roads consist of a graded and compacted roadbed usually created from the parent soil-material. The rolling wheels of the vehicles impart a force to the surface that pulverizes the roadbed material and ejects particles from the shearing force as well as by the turbulent vehicle wakes (Nicholson et al., 1989). Studies have found that dust emission rates depend on the fine particle content of the road (Cowherd et al., 1990; MRI, 2001), soil moisture content, vehicle speed (Nicholson et al., 1989; Etyemezian et al., 2003a, b), and vehicle weight (US EPA, 1996, 2003; MRI, 2001).

As part of a study to understand the contributions of military testing and training activities to regional particulate matter in the western US (Gillies et al., 2002), a study was undertaken to measure unpaved road dust emissions from wheeled vehicles and characterize their wakes and dust injection heights. These tests were carried out at Ft. Bliss, TX. A mix of civilian and military vehicles covering a substantial range of weights,

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length and width dimensions, and number of wheels were used to understand how these properties relate to emissions of dust, specifically the PM_{10} component. The results of these measurements are compared with modeled results using earlier (US EPA, 1996) and more recent (US EPA, 2003) estimation methods.

2. Experimental methods

From 11–24 April 2002, unpaved road emission flux experiments were conducted at Ft. Bliss, TX, using an upwind/downwind measurement technique similar to Gillies et al. (1999). Three towers were set up collinearly and perpendicular to a 1000 m section of unpaved road. The three towers were all downwind of the road at distances of 7, 50, and 100 m. A schematic diagram of the tower monitoring system is shown in Fig. 1. Each downwind tower was instrumented with four DustTraks (Model 8520, TSI Inc., St. Paul, MN) configured to measure PM_{10} that were spaced logarithmically (Fig. 1) in the vertical direction. The DustTrak is a portable, battery-operated, laser-photometer that uses light scattering technology to determine mass concentration in real-time.

The tower at 7 m downwind (DT_1) had measurement positions at 0.76, 1.28, 2.66, and 5.18 m above ground level (AGL). The second tower (DT_2) at 50 m had measurement positions at 1.25, 2.6, 5.7, and 12.2 m AGL. The third tower at 100 m (DT_3) had the same measurement positions as DT_2 with an additional sampling location at 0.4 m AGL. Five anemometers, one wind vane, and one temperature probe were mounted on DT_3 in order to characterize the local meteorological conditions. Dust concentration and meteorological data were collected and stored on PCs located at each tower running a custom-designed LabView data acquisition program.

Road dust PM_{10} emissions were created by having a test vehicle travel back and forth along the roadway for a number of passes. The test vehicles traveled at set speeds of 16, 24, 32, 40, 48, 56, 64, 72, and 81 km h^{-1}. After two passes at the same speed, the vehicle speed was increased incrementally to the maximum and then decreased incrementally to the slowest speed. Vehicles paused between passes to allow a plume to move by all three towers. Test vehicle descriptions and physical characteristics are presented in Table 1.

A three-dimensional sonic anemometer (Applied Technologies Inc., Boulder, CO) was used to assess the magnitude of the turbulent wake behind vehicles and assess its effect on the initial distribution of the vehicle-generated dust plume. The tests were carried out at night because under stable atmospheric conditions, the turbulence generated by a vehicle can be more readily identified with a sonic anemometer than when the atmosphere is unstable and the background turbulence level is much higher. Three vehicles with considerably different profiles were tested: a Dodge Neon, a 1979 GMC van, and a GMC C5500 truck. To facilitate measurement of the vehicle wake, tower DT_1 was moved to the edge of the roadway prior to testing. The sonic anemometer was mounted on the tower at a height of 0.9 m AGL and extended a further 1.1 m towards the road on a boom arm. Each vehicle was driven past the tower once heading north and once heading south at 16,

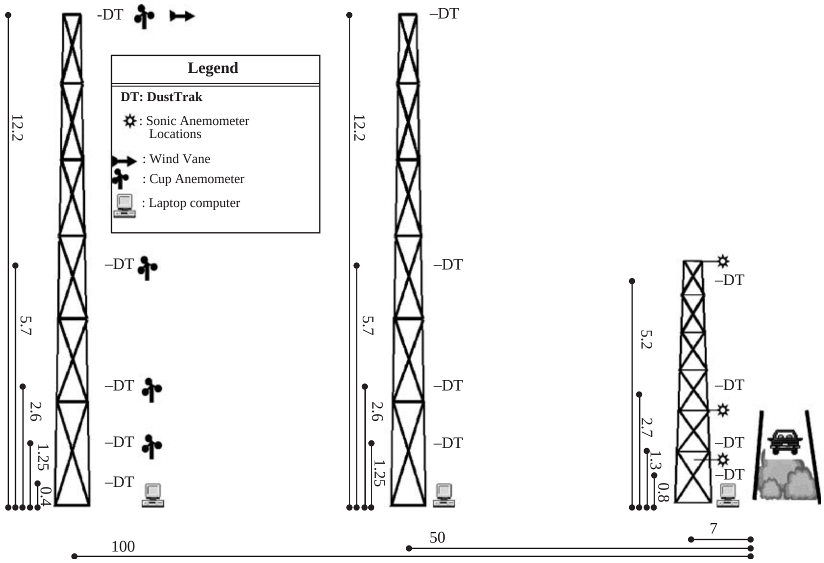

Fig. 1. Schematic diagram of the 3-tower dust monitoring system. Units are shown in meters.

Table 1
Test vehicle characteristics

Vehicle specifications							
Vehicle type	Model year	Weight (kg)	Length (m)	Width (m)	Height (m)	Under-carriage area (m^2)	Number of wheels
Dodge neon	2002	1176	4.34	1.71	1.42	7.43	4
Dodge caravan	2002	1759	5.09	1.83	1.76	9.31	4
Ford taurus	2002	1516	5.08	1.83	1.41	9.30	4
GMC G20 van	1979	3100	5.1	1.85	2.00	9.44	4
HMMWV	n/a[a]	2445	4.57	2.16	1.83	9.87	4
GMC C5500	1999	5227	9.04	2.29	3.30	20.67	6
M977 HEMTT	n/a	17727	9.96	2.44	2.85	24.29	8
M923A2 (5-ton)	n/a	14318	8.74	2.44	3.07	21.32	6
M1078 LMTV	n/a	8060	6.43	2.44	2.69	15.69	4

[a]n/a: data not available.

32, 48, and 64 km h^{-1} for a total of eight passes per vehicle. After all three vehicles completed this cycle the sonic anemometer was moved to 2.4 m and then to 5.5 m AGL. The driving pattern (three vehicles, four speeds, two passes at each speed) was repeated at each of those two heights.

Bulk surface samples were collected at the beginning and end of the testing to determine the percent silt content of the road material following the method of Cowherd et al. (1990).

3. Results

PM_{10} emission fluxes were calculated for each downwind tower using the assumption that each DustTrak represented the PM_{10} concentration over a height that spanned half the distance to the next lowest monitor to half the distance to the next highest monitor (Fig. 1). The time series of PM_{10} concentrations were examined for each DustTrak at each location. Each peak in concentration was associated with an individual vehicle pass and each pass was assigned a start and stop time. The emissions factor per vehicle pass for each downwind tower was calculated using the sum of the 1-s PM_{10} fluxes with the equation

$$\mathrm{EF} = \sum_{\text{start time of peak}}^{\text{end time of peak}} \left\{ \cos(\theta) \sum_{i=1}^{4} u_i C_i \Delta z_i \Delta t_i \right\}, \tag{1}$$

where the outer summation is over the period of plume impact, EF is the estimated emissions factor of PM in grams per vehicle kilometer traveled (g-PM vkt^{-1}), θ is the angle of the 5-min. average wind direction relative to the flux plane, i is monitor position on the tower, u_i is the 5-min. average wind speed (m s^{-1}) over the height interval represented by the ith monitor, C_i is 1-s PM concentration (mg m^{-3}) as measured by the ith monitor, Δz(m) is the vertical interval represented by the ith monitor, and Δt (s) is the duration that the plume impacts the tower. For all emission factor calculations only the concentration data associated with wind approach angles of $\leqslant 45^\circ$ with respect to the tower line were used. Winds speeds and directions measured on the third downwind tower were assumed to be representative of the speeds at the other towers.

For the period of the testing the moisture content of the roadbed material, expressed as a percent difference between the weight of samples before and after oven drying was <0.5%. The silt content of the road changed from 4% to 7%.

Examples of the emission of PM_{10} per vehicle kilometer traveled as a function of vehicle speed (E_s, g-PM_{10} vkt^{-1}) for the light (<4000 kg) and the heavier (>4000 kg) vehicles are shown in Fig. 2a and b. The emission factor value for each speed category is based on the average from multiple passes for all three towers and the error bars represent the standard deviation of the three-tower average. From the relationships depicted in these figures it is apparent that vehicle speed and size play important roles in the magnitude of the emissions.

The results of the nighttime wake turbulence tests are shown in Fig. 3, which illustrates that the size of the turbulent wake behind a vehicle increases with physical size. The GMC C5500 creates a measurable wake up to a height of approximately 6 m. The GMC G20 wake only goes up to about 2.5 m. The Dodge Neon is apparently too small to cause a measurable wake, even at the minimum anemometer height setting of 0.9 m. The dotted line drawn for the compact car is based on a linear fit of the wake heights of the box truck and the cargo van vs. their physical heights (3.2 and 2.0 m, respectively). The height of the compact car at the rear of the vehicle is 1 m giving an approximate wake height of 1.7 m according to this regression (see Fig. 4). The wake height is important because it gives an indication

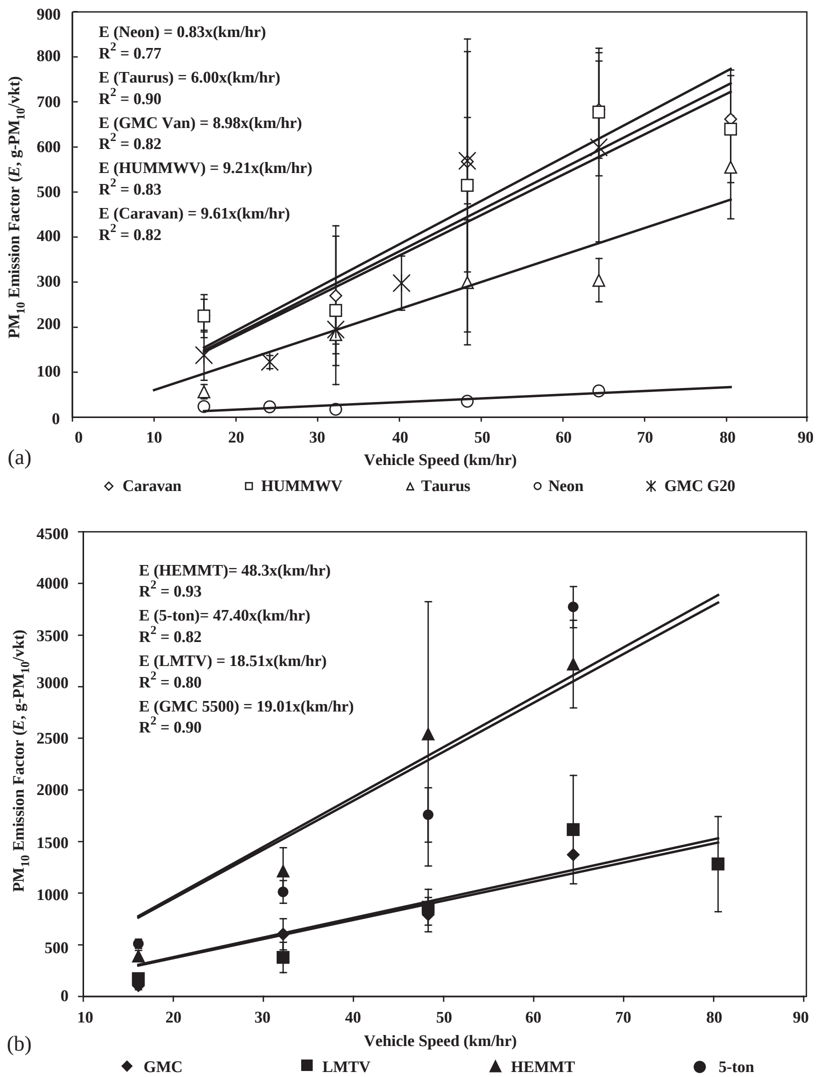

Fig. 2. (a) Emission factors as a function of speed for vehicles <4 000 kg. (b) Emission factors as a function of speed for vehicles >4 000 kg. Error bars represent the standard deviation of the mean value for that speed based on multiple vehicle passes, and for all three towers combined.

of the height to which the dust plume is mixed behind the vehicle. As a first approximation, we may assume that the dust emitted behind a vehicle is well-mixed up to the height of the turbulent wake (approximately 1.7 times the height of the vehicle). This "injection" height may play an important role when determining the fraction of particulate matter that deposits close to the road vs. the fraction that is transportable; as a rule, the further a particle is from the ground initially, the less likely it is to deposit in a given period of time.

The approximation for the injection height suggested here is simplistic. This height is likely to depend on a number of parameters, most notably the atmospheric stability, the shape of the vehicle, and the angle of the ambient wind with respect to the direction of vehicle travel. The analysis here is based on data obtained under stable conditions. Under unstable conditions, turbulent mixing is not inhibited by buoyancy forces and therefore, it is likely that the effective injection height will be larger because of this. For the present purpose, it is sufficient to note that the height to which a dust plume is thoroughly mixed in the wake of a vehicle is dependent on the size of the vehicle, increasing in a roughly linear relationship with its height.

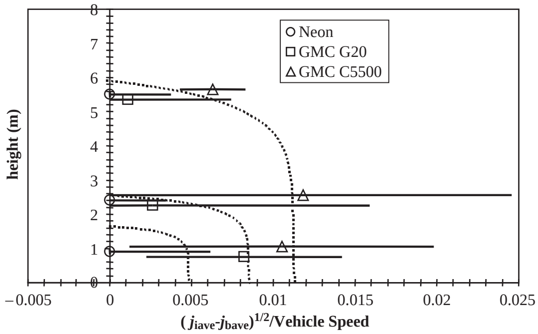

Fig. 3. Turbulence generated vs. height above ground for three vehicle types. The x-axis represents the difference between the vertical component of vehicle turbulence and the background fluctuations in vertical velocity normalized by the speed of the vehicle. The terms j_{iave} and j_{bave} represent the average value of the squared difference between the average vertical velocity component and its fluctuating component during the presence and absence (i.e., background), respectively of the vehicle wake. The horizontal lines represent the background turbulence standard deviation. The dotted lines represent hand drawn curves to fit these data. The dotted line corresponding to the compact car was drawn based on the assumption that the height of the wake plume is proportional to the height of the vehicle (see Fig. 4).

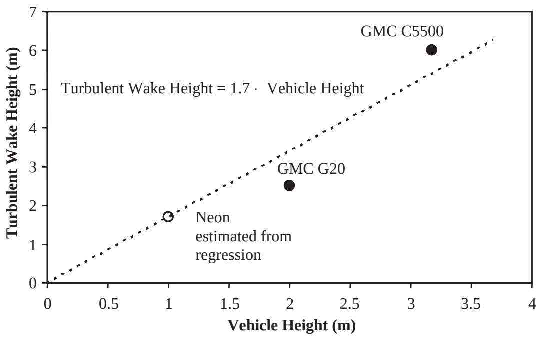

Fig. 4. Estimate of turbulent wake height vs. physical vehicle height. The dotted line represents a zero-intercept regression on the data from the GMC G20 and GMC C5500. The hollow circle is an estimate of the wake height for the Neon based on the regression.

4. Discussion

Fig. 2 demonstrates that vehicle speed is an important factor with respect to unpaved roadway PM_{10} emissions for the tested vehicles. The effect of speed on emissions is linear and relatively invariant with vehicle type as shown in Fig. 5. This figure shows the emissions normalized to the fastest speed obtained for each test vehicle. Regardless of test vehicle type the emissions from the roadway increase at a constant rate with increasing speed. MRI (2001) reported for other

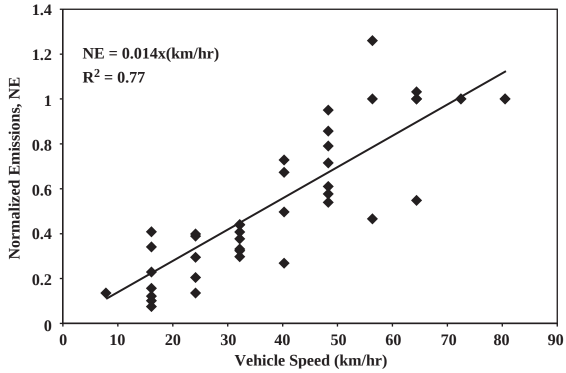

Fig. 5. The effect of vehicle speed on emissions independent of vehicle weight.

unpaved roadways that emissions vary with speed raised to a power typically between 1 and 1.5. This suggests that physical characteristics of the vehicles such as shape and number of tires and tread pattern may have only a minor influence on the emissions. There is, however, a discernable effect of vehicle weight on the emissions from the unpaved road. The slopes of the emission factors as a function of speed relationships, from the regression-derived equations presented in Fig. 2 are plotted against the weights of the vehicles in Fig. 6.

In earlier emission factor estimation methodologies the effect of vehicle weight on PM_{10} emissions was treated as a power function with the weight being raised to the 0.7 or 0.45 power. The Ft. Bliss data show a strong linear relationship between weight and emissions (Fig. 6). The vehicle undercarriage area and the number of wheels have weak and no discernable relationships with emission factors, respectively, as indicated by low correlation coefficients.

In comparing the measured emission factors from the vehicles tested at Ft. Bliss with other estimation methods several differences are clear. Using a US EPA method published in AP-42 in 1996 (US EPA, 1996) it was observed that on average this method under-predicted the emissions from each test vehicle by a factor of 2.4 ($\pm$0.9) for roads of 4% silt content and 1.4 ($\pm$0.5) for roads of 7% silt content over the range of vehicle speeds tested. More recent estimation methods offer two options for public roads and a separate method for industrial roads. Comparing the measured emission factors at Ft. Bliss with the US EPA (2003) emission model for vehicles traveling on publicly accessible roads, dominated by light duty vehicles, which is: E (lb- PM_{10} per vehicle mile traveled) $= [1.8(s/12)^1(S/30)^{0.5}/(M/0.5)^{0.2}]\text{-}C$, where s represents silt content (%), S is mean vehicle speed (mph), and M is surface moisture content (%), C is the emission factor for 1980s vehicle fleet exhaust, brake wear and tire wear, the Ft. Bliss emissions for individual vehicles are under-predicted

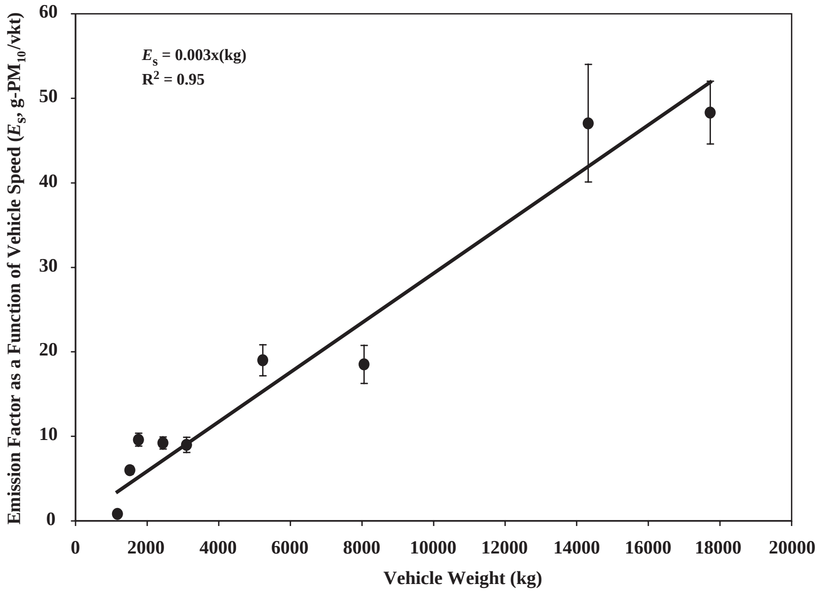

Fig. 6. The slope estimates from the relationship between emission factor and vehicle speed plotted as a function of vehicle weight. The error bars represent the standard error of E_s.

due to the difference between speed expressed as a power function in the model and the linear dependence observed in the data, except for the lightest vehicle (Dodge Neon).

The major difference between the modeled and measured values is caused by the observed effect of vehicle weight on emissions (Fig. 6). For the Ft. Bliss data the model over-predicts emissions for vehicles $<\sim$1176 kg for a 4% and 7% silt content and under-predicts for heavier vehicles. For lighter vehicles ($\sim$1176 kg) the measured to predicted value is ~0.4 to ~0.2 for silt contents of 4% and 7%, respectively. At the heavy end (~18 000 kg) the measured emissions are ~24 to ~14 times greater than the modeled values, for 4% and 7% silt contents, respectively.

The movement of the heavy military vehicles at Ft. Bliss can be compared to the movement of industrial vehicles on haul roads. For this case the US EPA (2003) proposed the emission model: E (lb-PM_{10}vmt^{-1}) $= 1.5(s/12)^{0.9}(W/3)^{0.45}$ where W represents the mean vehicle weight in tons. Using the industrial road model neglects the speed effect that was observed in the Ft. Bliss data. The result is that the model over-predicts emissions for speeds less than ~11 km h^{-1} for 4% silt content roads and ~21 km h^{-1} for 7% silt content roads and under-predicts emissions at faster speeds. At ~80 km h^{-1} the model under-predicts heavy vehicle ($>$5000 kg) emissions by a factor of ~6 to ~9 times for 7% and 4% silt contents, respectively.

The effect of injection height on emissions was assessed using the ISC3 model (US EPA, 1995) to model changes in the concentration profile downwind of a ground-level source (Etyemezian et al., 2003c). The initial dispersion parameter (σ_z) was assumed to equal one-half of the injection height. The modeling results indicated that the injection height for the dust plume is least important under unstable atmospheric conditions, and most important under stable atmospheric conditions. This is because under unstable conditions, the dust plume is lofted up high quickly anyway, more or less regardless of the starting height. Under stable conditions, the extent of vertical mixing is retarded by buoyancy, and a lower injection height allows the particles to be closer to the ground for a longer period of time, thereby enhancing deposition.

Comparison of downwind removal rates indicates that for unstable conditions, the injection height has a non-trivial, but small effect. For very stable conditions, the injection height has a large effect, approaching a factor of two difference in estimated removal rates at 1000 m downwind of the source. The difference between the removal of particles for two plumes with different injection heights is confined to the near-source region. That is, the difference does not continue to grow past about 1000 m downwind. With this in mind, and noting, as before that most unpaved road dust emissions are likely to occur in the daytime, when conditions are neutral to unstable, the effect of the injection height is

secondary to the uncertainties associated with deposition velocities and dispersion parameters.

5. Conclusions

As part of a larger study to assess the contributions of military training and testing to fugitive dust levels emission factors for a range of vehicles types for unpaved roads were measured at Ft. Bliss, TX in April 2002. A downwind array of three instrumented towers was used to measure the mass concentration of dust in the advecting plumes created by the vehicles traveling on the roadway. The magnitude of the emissions was controlled primarily by vehicle speed and vehicle weight, both of which had linear effects on the emissions. This suggests that emissions are linearly dependent on a vehicle's momentum. Other physical characteristics of the vehicles (e.g., # wheels, undercarriage, area, height) did not appear to heavily influence the emissions. The effect of the injection height of the dust as a function of vehicle wake size was found to be minimal in neutral and stable conditions, becoming important only in very stable conditions.

In comparison with US EPA AP-42 (1996, 2003) methodologies for determining contributions of fugitive dust from unpaved roads at Ft. Bliss, application of AP-42 emission models would generally under-predict the measured emissions for vehicles other than passenger cars by a factor between 2 and 24, depending on vehicle weight and speed. For passenger cars under ~1200 kg the AP-42 methods over predict emissions. An important question still remains to be resolved, which is "what is the eventual fate of these emissions"? There is still considerable discrepancy between the inventory-derived amounts of unpaved road dust contributions and the amount of mineral-type fugitive dust observed in the ambient air.

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